Ecological Dynamics of Wetlands at Lisbon Bottom, Big Muddy National Fish and Wildlife Refuge, Missouri

By Duane C. Chapman, Ellen A. Ehrhardt, James F. Fairchild,
Robert B. Jacobson, Barry C. Poulton, and Linda C. Sappington
U.S. Geological Survey, 4200 New Haven Road, Columbia, MO 65201

Brian P. Kelly, U.S. Geological Survey, 401 NW Capital Drive,
Lee's Summit, MO 64086

William R. Mabee, Missouri Department of Conservation, Resource
Science Center, 1110 S. College Avenue, Columbia, MO 65201

Final Report to the U.S. Fish and Wildlife Service
Big Muddy National Fish and Wildlife Refuge, Columbia, MO
December 2002, revised December 2003

Open-File Report 2004-1036

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
Gale A. Norton, Secretary

U.S. Geological Survey
Charles G. Groat, Director

U.S. Geological Survey, Reston, Virginia: 2004
Revised and reprinted: 2003

For more information about the USGS and its products:
Telephone: 1-888-ASK-USGS
World Wide Web: http://www.usgs.gov/

Ecological Dynamics of Wetlands at Lisbon Bottom, Big Muddy National Fish and Wildlife Refuge, Missouri

By Duane C. Chapman, Ellen A. Ehrhardt, James F. Fairchild,
Robert B. Jacobson, Barry C. Poulton, and Linda C. Sappington
U.S. Geological Survey, 4200 New Haven Road, Columbia, MO 65201

Brian P. Kelly, U.S. Geological Survey, 401 NW Capital Drive,
Lee's Summit, MO 64086

William R. Mabee, Missouri Department of Conservation, Resource
Science Center, 1110 S. College Avenue, Columbia, MO 65201

Chapter 2

Limnology of Lisbon Bottom Wetlands

by Duane C. Chapman, James F. Fairchild, and Ellen A. Ehrhardt

Final Report to the U.S. Fish and Wildlife Service
Big Muddy National Fish and Wildlife Refuge, Columbia, MO
December 2002, revised December 2003

Open-File Report 2004-1036

U.S. Department of the Interior
U.S. Geological Survey

Chapter 2. Limnology of Lisbon Bottom Wetlands

Duane C. Chapman, James F. Fairchild, and Ellen A. Ehrhardt

Abstract

In this study we examined the limnology of a continuum of wetland types at Lisbon Bottom in order to examine the physical, chemical, and biological characteristics of wetlands in relation to timing, source, and duration of flooding. We examined the relationship between inundation and water quality of thirteen Lisbon Bottom wetlands with varying degrees of permanence and water sources from late March through July 1999. Temperature and water quality of the Missouri River and of the Lisbon Chute were also measured. Growth of fathead minnows (*Pimephales promelas*) in cages was measured as an estimate of secondary productivity.

We found that nutrient availability, chlorophyll *a* concentrations, and fathead minnow growth were higher in wetlands that were influenced by the river than in wetlands that were mostly stream-influenced. Aquatic macrophytes in stream-influenced wetlands removed available nutrients from the water column after the water warmed in late April, resulting in low phytoplankton productivity and thus lower zooplankton productivity. River-influenced wetlands received large inputs of nutrients from the river during floods. Chlorophyll *a* concentrations increased in response to the nutrient inputs as soon as turbidity from the flood event diminished. In contrast to the stream-influenced wetlands, productivity in the river-influenced wetlands is likely driven by phytoplankton rather than macrophytes. Although moist-soil and inundation-tolerant vegetation such as cocklebur, cottonwoods and willows were often inundated by high water in the wetlands, obligate aquatic macrophytes were generally not present in the river-influenced wetlands because of high turbidity, sedimentation, water depth fluctuations, and sometimes short duration of inundation. Nutrient concentrations in the river-influenced wetlands peaked after floods, but nitrogen dissipated rapidly after flooding subsided.

Aquatic habitat conditions were very dynamic. Flooding events modified the topography of the wetlands during this study and between earlier studies and this one. Wetlands that were backflooded by the river were connected to the river more often than topflooded scours, but with less energy. Temperature stratification in deep topflooded scours was completely destroyed by flood events, but this did not happen in the deep backflooding scour. Dissolved oxygen concentrations in the hypolimnia of scours rapidly decreased after stratification.

Introduction

Lisbon Bottom, a unit of the Big Muddy National Fish and Wildlife Refuge, is located within a sharp bend of the Lower Missouri River south of Glasgow, Missouri. The flood of 1993 breached agricultural levees around Lisbon Bottom, thereby re-establishing a dynamic connection to the Missouri River. Repeated flooding of the bottom between 1993 and 1999 changed this previously agricultural area into a diverse ecosystem, with unvegetated sandy areas, large areas dominated by cocklebur, dense stands of young cottonwoods and willow, and a variety of temporary and permanent wetlands.

The hydrology of Lisbon Bottom is extremely dynamic and is dependent on the sources and timing of water in the flood plain (see Chapter 1). The various sources of water entering the flood plain (river, groundwater, rainfall, and tributary inflow) vary widely in water quality characteristics; however, there have been relatively few studies to examine the limnology and water quality in the Lower Missouri River flood plain. Much of the existing data for the Lower River is contained within the databases of the U.S. Geological Survey's National Stream Quality Accounting Network (NASQAN) Program. However, this data is measured only on a monthly basis at a few, fixed sites on the river and critical water quality parameters related to aquatic structure and function (for example, particulate organic carbon and chlorophyll) are not measured (Blevins and Fairchild, 2001). Heitmeyer and others (1991) examined the water quality and hydrologic relationships among a continuum of wetland types in a managed wetland complex and found distinct differences in wetland type depending on depth and vegetative characteristics. Knowlton and Jones (1997) found that the river had a strong influence on scour holes connected to the river but that non-connected scours developed a separate limnological trajectory due to *in situ* biological processes that served to reduce concentrations of dissolved and total nutrients. Knowlton and Jones (2000) examined the suspended solids, nutrients, and chlorophyll of the Lower Missouri River over a 5-yr period and found strong, seasonal relationships in these components due to hydrologic and seasonal variation.

In this study we examined the limnology of a continuum of wetland types at Lisbon Bottom in order to examine the physical, chemical, and biological characteristics of wetlands in relation to timing, source, and duration of flooding. These data are provided as a basis for an understanding of the complexity of wetland structure and function in a dynamic ecosystem such as the Lower Missouri River. Such knowledge is essential to biologists involved in purchasing and managing habitats in the Lower Missouri River ecosystem.

Methods

Limnological studies were conducted in 13 wetlands from late March through July, 1999. A description of wetland characteristics (type, description, primary water source, and relative degree of permanence) is provided in Table 2-1. Locations of the wetlands are depicted in figure 2-1.

Wetland Water Depth and Periods of Inundation

Water elevations in individual wetlands were determined empirically using wooden staff gages installed early in the study. Staff gages were installed in the deepest portion of the shallow wetlands. In the deeper scour wetlands (also known as evorsions or blew holes) the gages were installed where they would be legible from the shore. Water elevation (or presence-absence of water) was recorded from the staff gages on each visit to a wetland, regardless of the purpose of the visit (bird, fish, invertebrate or limnology portions of the study). These data were later converted to water-surface elevations by surveying in the gages relative to established elevation benchmarks. Very high water levels impeded collection of staff gage data, because of inaccessibility to walkers or, in the case of Wetland 26, complete inundation of the staff gage by several feet of water. Staff gage data was collected at most wetlands by canoe during the flood on April 30. These data should approximate the highest water levels experienced during the study. Beavers repeatedly destroyed the staff gages in some wetlands, resulting in some periods without water elevation data. Wooden staff gages were eventually replaced with metal gages in those wetlands, which eliminated this problem.

Water Quality

Water samples were taken twice weekly, from 3/30/99 to 6/15/99, for analysis of pH, dissolved oxygen (DO), turbidity, conductivity, hardness, alkalinity, chlorophyll *a*, particulate organic carbon (POC), total nitrogen and total phosphorus. Sampling was performed as a part of the bird observation portion of this study; the observers sampled water after the allotted observation period of each wetland. Thus, samples were taken during the morning, but the timing of sampling varied depending on when the observer completed the observation period and moved to the next wetland.

Duplicate 250 mL samples were taken by submersing the sample bottles in the wetland and filling them to the top. Air space in the container was eliminated or minimized. Though many of the samples were taken by wading, care was taken to acquire the sample without disturbing sediment and artificially increasing turbidity values. The temperature of the wetland was taken using a field thermometer and recorded on field data sheets. The bottles were iced and transported to CERC. One sample was used for DO, pH, turbidity, conductivity, hardness, and alkalinity analysis on the day of sample collection. A portion of the other duplicate water sample was filtered and extracted on the day of collection for chlorophyll analysis. The remainder of that bottle was frozen and later analyzed for nutrients.

Dissolved oxygen was determined with a YSI® model 57 dissolved oxygen meter and conductivity was determined using a YSI® S-C-T meter. Orion® EA 940 meters with glass gel electrodes were used to measure pH and alkalinity. Total nitrogen and total phosphorus were analyzed using a Technicon® Autoanalyzer. Particulate organic carbon was measured using a Coulometrics Model 5020 Carbon Analzyer. Chlorophyll *a* was determined by acetone extraction and use of a Turner Designs® 10-AU Flourometer.

In addition, *in situ* temperature loggers (Onset Corporation® tidbit loggers) were installed in selected wetlands in order to measure temporal changes in temperature and any subsequent stratification. Loggers were suspended 10 cm below a float, which was anchored in the deepest portion of the wetland. The float was attached to the anchor with sufficient cord that the float was not submerged during flooding. In the deepest scour wetlands (Wetlands 4, 5, and 26) a second logger was attached just above the anchor to record the temperature at the bottom. Additionally, three loggers were installed in the chute (one near the upstream end of the chute, one near the center and one just below the grade control structure near the lower end of the chute) and one logger was installed in the Missouri River on the outside bend of the river southeast of Lisbon Bottom.

Temperature and dissolved oxygen profiles (half-meter intervals) were taken weekly using a boat at two points in each of the deep scours using a YSI® model 57 temperature/DO meter.

A two-tailed Student's *t* test was used to compare water quality parameters between stream-influenced and river-influenced wetlands.

Fathead Minnow Growth Study

Survival and growth of fathead minnows were studied in the thirteen wetlands as an integrator of the combined influence of limnological characteristics (for example, temperature, dissolved oxygen, nutrients, and chlorophyll). Studies were conducted in small polyethylene cages that were deployed over a 20-d period (5/27 through 6/16). Fathead minnows were acquired from Aquatic Biosystems, Inc. (Fort Collins, CO) to assure low variance in starting weight. Initial fish weight was determined by taking the mean of 20 fish (29.10 mg dry weight; 144.1 mg wet weight). Both dry and wet weights were determined, but dry weights were used for the analysis because the weight of water on the surface of the fish can bias measurements of small fish. Growth in this study was defined as mean weight of surviving fish at the end of the study minus the mean starting weight.

Two 15 x 15 x 15 cm cages were installed per wetland, with 5 fish per cage. The cages were attached to the bottom side of a wooden float, so that they were completely submerged but just below the surface at all times, and at least partially shaded. Water temperature at the cages was logged every half hour using tidbit loggers (Onset® Corporation). Fish were transported using plastic bags with oxygen and installed in the cages on site. During the fathead minnow study, dissolved oxygen was measured weekly in the morning at the cages using a YSI® model 57 dissolved oxygen meter.

A two-tailed Student's *t* test was used to compare fish growth and water quality parameters between stream- and river-influenced wetlands, and between topflooding and backflooding wetlands.

Results and Discussion

Wetland Water Depth and Periods of Inundation

As is normal for spring and early summer on the Missouri River, river stage varied greatly during the period of the study. Portions of Lisbon Bottom that were completely dry for most of the period of the study were occasionally inundated with several feet of water (fig. 2-2A and 2-2B). Water surface elevation of selected wetlands by date is displayed in figure 2-3. With the exception of measurements made by canoe on 4/30/99, these data do not capture the periods of highest flooding from the river, but the figures show that most wetlands were strongly influenced by flooding from the river. Some of the wetlands were dry until they were filled by rain in early April (fig 2-2B), but almost all wetlands showed marked increase in surface elevation after the flood on April 16. The exceptions to this rule were Wetland 11 and Wetland 12. Wetland 11 had little relief and thus retained little water after the flood, although continuous flow from Lay Creek kept the wetland hydrated. Wetland 12 was separated from the flood plain by a secondary levee and thus was never flooded. Wetlands 4 and 5 were strongly connected to the river on 4/16/02 via a crevasse that passes through the upstream levee when river stage at river mile 218.2 exceeded 185.5 m above mean sea level (fig. 2-4). This river stage corresponds to approximately 25.5 ft on the Boonville gage. Stage data from Boonville is not exactly comparable to the Lisbon area because of the influence of the Lamine and other tributaries and because of differences in flood-plain morphology. Also, continued erosion of the crevasse or blockage by sediment and woody debris may have affected the water stage at which the crevasse provided water to these wetlands on later dates. However, during this study, Wetlands 4 and 5 did experience strong mixing at other times when the

Boonville river stage exceeded 25.5 ft. Other topflooding wetlands were supplied by water that passed through Wetlands 4 and 5 or entered directly from the chute (note strong water surface elevation changes on this date for all wetlands in fig 2-3). During this study there were three high-water events that caused topflooding. These events peaked on 4/18/99, 4/30/99 and 5/6/99 (fig. 2-4).

During the highest flooding periods, some water overflowed the chute and moved diagonally across the southern portion of the bottom at high velocity, returning to the river at the exit scour. Although it was impossible to view these high flow occurrences for safety reasons, the high flows were evidenced by at least 10 m head-cutting horizontally from the scour toward the chute (the head cut crossed an ATV trail that we had previously used as a path) and by cottonwood saplings of up to an inch in diameter that were pushed over and laying horizontally on the ground after the flood. In other portions of the bottom, water velocities during flooding were generally low, except possibly in the sections where Wetlands 4 and 5 were being fed by the river through the crevasse. During a comparable flood (283,000 cfs) in 1998, velocities recorded in the dense willows of the NE quadrant of Lisbon Bottom were consistently < 30 cm/sec (Robb Jacobson, personal communication). During floods, it was impossible to reach the chute or wetlands in the southern and western portions of the bottom because of safety concerns due to high and fast water. However sampling was continued as much as possible in the northern and eastern sections of the bottom, by wading and with the use of a canoe.

Wetland 26 was connected to the river via backflooding at about 20.5 ft (Boonville gage). Backflooding provided water to Wetland 22 and much of the southern bottom at river stages lower than that required for topflooding. Thus, these areas were directly connected to the river during more of this study than the topflooding wetlands.

Flooding events visibly modified the flood plain during the course of the study, and between this study and an earlier study in 1997. A single minor flood event deposited up to 10 cm of unconsolidated sediment over a large area of the southern bottom (fig. 2-5) and scouring occurred in other areas. For example, over the course of the study, Wetland 8 experienced scouring in the upper end and sedimentation in the lower end. Wetland 5 is significantly shallower than it was two years earlier as evidenced by approximately 20 cm of sedimentation around staff gages left in place from an earlier project. These fine sediments can provide a clay barrier to water movement (Chapter 1). Thus, while deposition of fine sediments tends to fill the deep scour wetlands and eventually limit their lifespan, it may in the short term provide water level stability and prevent drying of scours when the river stage and water table are low.

Staff gage data from Wetlands 11 and 12 in 1999 indicate that these wetlands were the most stable in surface elevation (fig 2-3). This contrasts with surface elevation data collected in 2000 using automatic recorders, which indicate a higher frequency and degree of surface elevation change in stream-fed valley-wall wetlands (Chapter 1). These differences may be more related to the methods than to differences in hydrology. Staff gage data were taken only when personnel were present at the site in 1999, and during very high flows, measurement was difficult. Thus the 1999 data may not have captured all of the variability of these wetlands. However, the 1999 staff gage data from these wetlands are significant in that minimum water surface elevations were extremely stable compared to the other wetlands. Stream flows during non-flooding periods maintained hydration and inundation of these wetlands much longer than other shallow wetlands elsewhere on the Lisbon flood plain. This duration of inundation was important in providing appropriate conditions for aquatic macrophyte growth, and in providing habitat for invertebrates and waterfowl (Chapters 4 and 6).

Water Quality

Water qualities of the river and chute were almost identical. Turbidity and total nitrogen concentrations were higher (fig. 2-6 and fig. 2-7), and conductivity lower (fig 2-8), during periods of high flow. Hardness and alkalinity followed the pattern of conductivity. The temperature logger in the river was lost during the study, but data collected after the study was completed, indicate that the chute, with its higher surface area to volume ratio, is apparently more affected by changes in ambient temperature and solar radiation than the river, especially at the lower end. The river had much more thermal inertia and did not vary as much in temperature (fig. 2-9). It should be noted that the fig. 2-9 temperature data are taken from a period during which the river stage was comparable to the period prior to the spring flood, and that diurnal changes in temperature in the chute were not as pronounced during periods of higher flow (fig 2-10).

Based on the flooding regime experienced during the 1999 season, the studied wetlands were categorized loosely into topflooding, backflooding, stream- and runoff-influenced, and mixed influence groups based on the perceived major influences on each wetland. This classification is not perfect in its objectivity and distinctions are rough because all wetlands were at least somewhat mixed in influence and these influences cannot be exactly quantified by our data. However, we believe the distinctions based on our observations are adequate for making broad distinctions among these habitat types for the period examined. We considered Wetlands 4, 5, 8, 16, and 21 to be primarily topflooding wetlands, Wetlands 26 and 29 to be backflooding, and Wetlands 11 and 12 to be stream-influenced. The remainder of the wetlands were considered to have mixed or indeterminate influences. It should be noted that Wetlands 4, 5, and 8, which topflooded through a crevasse from the river probably flooded with a different frequency than Wetlands 16 and 21, which topflooded from the chute.

Of the 13 wetlands in which water quality was measured, nine (4, 5, 8, 9, 11, 12, 16, 22, and 26) held water at the beginning of the study. Wetlands 8 and 9 dried completely between the beginning of the study and the first flood.

The water quality of these wetlands was highly variable over time because of the degree of influence of flooding from the river and rain events on these small water bodies. For example, conductivities of river-influenced wetlands were lower than stream-influenced wetlands, until the river, which has a high conductivity, flooded the river-influenced wetlands on April 16 (fig. 2-11).

The overall mean pH of the wetlands was 8.2 (standard deviation 0.29). The pH varied from a low of 7.7 (Wetland 26 on the first day of the study) to 9.2 (Wetland 2 on 4/23/99). The most likely cause of pH changes and differences in these wetlands is photosynthesis. Since photosynthesis varies by time of day and degree of solar irradiance, the pH also changes similarly. Since it was impossible to sample all of the wetlands at once, or even at the same time for each wetland, pH data are useful primarily as an overall indicator.

Mean total phosphorus concentrations over the study were high, ranging from 181 to 408 µg/L in the topflooding and mixed influence wetlands, and over 300 µg/L in the backflooding and the stream-influenced wetlands and the river and chute (table 2-2). Mean N:P ratios for wetlands in this study ranged between 1.6 and

7.7. Nitrogen and phosphorus values measured in the river were similar to those measured by the U.S. Geological Survey in other years for this time period (Hauck and others, 1997).

Flooding by the river is clearly an important source of nutrients for these wetlands. The average total nitrogen concentration in the river and chute (2.7 mg/L) was nearly twice as high as that of the wetlands (1.5 mg/L). Following the 4/16/99 flood event, nitrogen concentrations in Wetland 5 changed from 2.0 mg/L to 4.3 mg/L and Wetland 4 concentrations changed from 1.2 mg/L to 1.9 mg/L. Nitrogen concentrations then rapidly decreased after the flood receded (fig. 2-12). This effect was stronger in shallow wetlands than deep wetlands. This rapid loss of nitrogen is frequently observed in shallow, lentic systems due to biological uptake by periphyton associated with sediments and detrital material (Knowlton and Jones, 1997; Flenniken, 2001). Denitrification, which occurs in carbon rich, anaerobic sediments, may have also been a factor. This study did not include an analysis of sediments and it is unclear whether sediments of shallow, temporarily flooded wetlands of Lisbon Bottom have the characteristics required for denitrification. Wetlands 11 and 12, which were the most strongly stream influenced, differed in nutrient dynamics from the rest of the wetlands. Wetland 12 did not receive a large pulse of nutrients during the flood and was the only site with submerged aquatic macrophytes; thus, nitrogen levels were much lower during the study. Wetland 11 was flooded briefly though deeply during the flood event (fig. 2-3), but since it did not retain water after the flood and it was strongly influenced by stream flows, by the time of our sampling, nitrogen concentrations were not markedly increased. Heavy rains in the watershed briefly raised nitrogen concentrations in the stream-influenced wetlands when water concentrations of nitrogen in the river-influenced wetlands were decreasing. Phosphorus concentrations were also generally higher in the river and chute than in the wetlands, although some wetlands would be considered very high in phosphorus.

POC values in this study (table 2-2) did not follow a clear pattern. In turbid systems, POC tends to correlate with turbidity, and in clear systems or systems with significant phytoplankton, POC will correlate with chlorophyll *a*. In these wetlands POC was likely influenced both by turbidity from the river and by subsequent algal production after river-born particulates precipitated. Because these two factors are negatively correlated, POC data were difficult to interpret.

Mean chlorophyll *a* concentrations, an important index of primary productivity, were high in river-influenced wetlands and low in stream-influenced wetlands and in the river and chute ($p < 0.01$; fig. 2-13). We expected turbidity to be highly correlated to nutrient concentrations in the wetlands, because suspended sediment should bear large quantities of nutrients. We found this only to be the case in the river samples ($r= 0.77$ for N and 0.73 for P correlations with river turbidity). This is probably because variables other than suspended sediment, such as phytoplankton, determine turbidity in the wetlands. Some wetlands, especially 10 and to a lesser extent 8, had generally lower chlorophyll *a* concentrations probably due to shading from trees.

Somewhat surprisingly, surface temperature did not vary according to wetland type, but all wetlands were warmer and had a higher daily variance in temperature than the river and chute (fig. 2-10).

Wetlands 4 and 5 were quick to stratify in the spring, with temperature and dissolved oxygen in the hypolimnia dropping quickly with depth (fig. 2-14). At times, the oxygenated epilimnion consisted of little more than the upper meter (fig. 2-14, 4/14/99 and 5/26/99). These wetlands were strongly influenced by flooding from the river, which destroyed the stratification (fig. 2-14, 4/30/99). However, stratification was quickly reestablished after the flooding events. This relationship is also shown by the temperature logger data. During flooding events, the temperature of the loggers at the top and the bottom of the wetlands are equivalent, and these values separate quickly after the flood event ends (fig. 2-15). Rain events also influenced stratification. Mixing occurred in Wetland 4 on 5/17/99 at a lower river stage than that at which the crevasse would have transferred water to it. Local rains in excess of 2” that occurred on this date (from USGS rain gage, see Chapter 1) apparently provided enough inflow to mix Wetland 4. There is a small intermittent stream (Buster Branch) that enters Wetland 4 on the northeast corner. Wetland 4 can also receive runoff via channels from Wetlands 3 and 6. Water quality measurements on 5/18/99 show an increase in nitrogen concentration but a decrease in conductivity and hardness. This is additional evidence that mixing from runoff was responsible, because inflows from the river would likely have increased rather than decreased conductivity and hardness.

Wetland 26 differed from Wetlands 4 and 5 in its pattern of stratification. This scour wetland is larger and deeper and thus has more thermal inertia. It was connected to the river more often than Wetlands 4 and 5, but being primarily backflooded as opposed to topflooded, water inflows were lower in energy. Thus, Wetland 26 did not stratify as strongly early in the observation period but once stratification was established, it remained stratified during flooding events (fig. 2-14).

Most small, permanent, nutrient-rich, lentic wetlands in Missouri have a high density of submerged and/or emergent aquatic macrophytes. Wetland 12 had high densities of macrophytes, but permanent Wetlands 4, 5, 16 and 26 did not. The absence of vegetation in these river-influenced permanent wetlands is presumably a result of the turbidity, scouring, sedimentation and severe water level fluctuations caused by repeated flooding by the river. Therefore, primary productivity in the river-influenced permanent wetlands is a function of phytoplankton production, which increased during periods of low inorganic turbidity. Sequestration of nutrients by macrophytes in Wetland 12 probably limits phytoplankton productivity in these wetlands, and primary productivity in these wetlands is primarily a function of macrophyte production. These differences are important in terms of zooplankton growth and thus to birds and fish which consume the zooplankton.

Most temporary and ephemeral wetlands were unvegetated basins, surrounded on their margins by smartweed, cocklebur, willows and small cottonwoods, which are considered to be moist-soil vegetation (Fredrickson and Taylor, 1982). Wetlands 9, 10, 11 and 22 differed. When not in flood by the river, Wetland 11 was very shallow and at times had perceptible current from Lay Creek. Wetland 11 was braided and highly vegetated with emergent macrophytes, primarily rice cutgrass with some perennial smartweed and spirea as well as needlerush, duck potato, spikerush, and American bullrush. Wetland 22, which in years prior to the study had remained more moist due to a combination of runoff through Wetland 11 and repeated backflooding through Wetland 26, had cattails, sedges, and bulrushes at the margins at the beginning of the study. However, emergent aquatic vegetation in Wetland 22 died during 1999, which was dryer than previous years. Portions of Wetland 9 and most of Wetland 10 were covered with dense growths of young cottonwoods and some willows, ranging 2–5 cm in diameter at the base.

All water quality data from the wetlands are found in Korschgen and others (ArcView-based spatial decision support system for the Lisbon Bottom Unit of the Big Muddy National Fish and Wildlife Refuge, unpub. data, 2001).

Fathead minnow growth study

Survival of fathead minnows was high in all wetlands except 9 and 10, which dried during the exposure period, and in Wetland 11, where survival was 50%. There was no apparent relationship between fathead minnow growth and whether wetlands were backflooded or topflooded. Fathead minnow growth was higher in river-influenced wetlands than stream-influenced wetlands ($p = 0.015$; fig. 2-16). This is most likely a combination effect of increased productivity due to the nutrients provided by flooding, and because phytoplankton (as opposed to macrophyte) productivity dominated the river-influenced wetlands. This illustrates the difference in nutrient pathways between the stream-influenced and vegetated wetlands and the river-influenced and unvegetated wetlands as described above. Fathead minnows feed mainly on algae and on crustacean zooplankton (Pflieger, 1997). Although the stream-influenced wetlands had very high densities of macroinvertebrates (Chapter 4) the caged fish were apparently unable to take advantage of the invertebrates as a food source.

Chlorophyll *a* was higher in the river-influenced wetlands during the *in situ* growth study ($p = 0.018$; table 2-3; fig. 2-16), as well as overall during the period of the study (fig. 2-13). However, chlorophyll *a* and growth were not highly correlated across all wetlands ($r = 0.319$). This could be because fish in the river-influenced wetlands were growing at their maximum rate well below the maximum chlorophyll *a* concentration, or other factors could be the cause. POC values, which are influenced by phytoplankton concentrations as well as turbidity, were also somewhat lower in stream-influenced wetlands ($p = 0.056$; table 2-3). Zooplankton density was low (fig. 2-16, table 2-4) in the stream-influenced wetlands, which may have influenced fathead growth, but zooplankton density was also low in Wetlands 4 and 22, where the fathead minnows had high growth. However, zooplankton density was more highly correlated to growth ($r = 0.722$) than was chlorophyll *a* concentration ($r = 0.369$). It also should be noted that although zooplankton density was very low in stream-influenced Wetland 12 during the three weeks of the caged fish growth study, zooplankton density in that wetland was reasonably high on average for the study as a whole (table 3-1).

Mean temperature overall during the fathead minnow study (from loggers) in wetlands that did not dry was 25.0 °C. Maximum temperature in the wetlands ranged from 32.5 to 35.9 °C. The loggers were co-located with the cages, so these temperatures reflect the temperatures to which the fish were exposed. Fathead minnow growth was not correlated to temperature ($r = -0.186$) or maximum temperature ($r = 0.167$) during the study, and these were not significantly different between wetland types. Negative correlation of growth with maximum temperature would have been expected if heat stress was a factor in some wetlands, but this did not occur and in fact the wetland with the highest maximum temperature had the highest growth. Some positive correlation of growth with temperature was expected, but this also did not occur. However, it must be noted that an equipment malfunction resulted in the loss of most of the temperature logger data from Wetlands 2, 11, 12, and 22. Since growth in Wetlands 11 and 12 was lower than that of the other wetlands, we re-examined the data using temperature data only from the three days at the end of the study period, in which data existed for those wetlands. During this period mean temperature in the stream-influenced wetlands was similar to the river-influenced wetlands (25.9 vs 26.0 °C; $p = 0.95$). Thus, there is no evidence that temperature differences were responsible for the lower fathead minnow growth rates in the stream-influenced wetlands, but the data are incomplete. Among only the river-influenced wetlands for which complete data was obtained, mean temperature was not correlated to growth (r = -0.123), but the mean temperature was not highly variable between wetlands.

Dissolved oxygen measured weekly in the morning at these wetlands was generally adequate for fathead minnows, with an overall average of 6.6 mg/L (fig. 2-17). However, on one date DO measured 2.4 mg/L at Wetland 5. All other measurements at Wetland 5 were over 7 mg/L. Overall survival at Wetland 5 was 80% (Fig. 2-16). Five mg/L DO is considered adequate for all fish (Piper and others, 1982) and fathead minnows are noted for their tolerance for low dissolved oxygen concentrations (Pflieger, 1997). The measured concentrations of dissolved oxygen, with the potential exception of the low value at Wetland 5, were not likely to be stressful. However, lower dissolved oxygen concentrations may have occurred earlier in the morning or on days when DO was not measured. Although an effort was made to sample these wetlands as early as possible in the morning, less accessible wetlands were sampled later in the day when photosynthesis had likely begun to raise oxygen concentrations. Because of this bias, we have not attempted to correlate measured dissolved oxygen concentration to growth.

Summary and Conclusions

The Missouri River is a primary source of nutrients for most of the wetlands. Nitrogen concentrations decreased rapidly either due to denitrification or uptake by terrestrial plants, especially in shallow wetlands. Wetlands that were river-influenced had much higher phytoplankton productivity than stream-influenced wetlands. Caged fish also grew faster in river-influenced wetlands. Zooplankton production in these wetlands was linked to phytoplankton concentrations that are in turn linked to nutrients from the river.

A crevasse through the levee at the northern end of Lisbon Bottom provided water and nutrients to wetlands that might otherwise have been inundated by the river for much less of the study or not at all. If continued river inundation of these northern wetlands is desired, the crevasse should be maintained and not plugged. There is a significant possibility that this crevasse could be plugged with river debris or it could possibly be plugged during levee maintenance operations. However, continued inundation by less than very large floods will allow continued sedimentation of Wetlands 4 and 5. The topflooding wetlands are less valuable to riverine fish (see Chapter 5) than wetlands on the downstream portion of the bottom, because they are connected to the river less often and for shorter periods, thus flooding periods are less likely to correspond with the time of year in which flood-plain spawners are actively spawning. Also, fish and their offspring spawned on the river bottom are more likely to be trapped in the topflooding wetlands because opportunities for egress to the river are more limited.

Topflooding Wetlands 4 and 5 stratified rapidly and had narrow oxic epilimnia, and hypolimnia that rapidly became anoxic. Stratification was destroyed during flood events and in Wetland 4 by runoff from a heavy rainfall, but was quickly reestablished. Wetland 26, a deeper, backflooding scour, did not completely lose stratification during flood events. Stratification patterns in these wetlands have important fishery implications (see Chapter 5). The river is continuing to modify these wetlands through sediment deposition in some areas and erosion of other areas. Further studies at this site will likely find shallow wetlands similar to those that we studied, but the specific wetlands may disappear or change in morphology. The future of the deeper scours is unclear. Wetland 5 changed from a relatively deep "permanent" scour in 1997 to a very shallow basin by the end of this study. The exit scour S-14 at the southern edge of Lisbon Bottom (not examined in the limnology portion of this study) was a deep scour in 1996 (Tibbs and Galat, 1997) but was very shallow over most of its area by 1999, and almost entirely filled in by mid-summer of 2000. Sedimentation of the deep scours occurs during flood events. Strong flows from a very large flood would likely be required to maintain or recreate these deep wetlands.

The river and chute were very similar in water quality, except for temperature. The chute was more variable in temperature than the river because of less thermal inertia and higher surface area to volume ratio. The chute was attractive to a diverse species assemblage of fish (Louise Mauldin, fisheries biologist, USFWS, unpub. data) and the river just downstream from the chute was occupied at times by pallid sturgeon (Aaron Delonay, ecologist, personal communication). Temperature regime could be a factor in the attractiveness of the chute to these fishes.

References

Blevins, D. and Fairchild, J.F., 2001, Applicability of NASQAN data for ecosystem assessments on the Missouri River: Hydrologic Processes, v. 15, p. 1347-1362.

Flenniken, L.A., 2001, The fate and effects of nitrogen and phosphorus enrichment in shallow ponds: Columbia, University of Missouri, M.Sc. thesis, 212 p.

Fredrickson, L.H. and Taylor, T.S., 1982, Management of seasonally flooded impoundments for wildlife: U.S. Fish and Wildlife Service Resource Publication 148, 30 p.

Hauck, H.S., Huber, L.G. and Nagel, C.D., 1997, Water resources data, Missouri water year 1996: U.S. Geological Survey Water-Data Report MO-96-1, 292 p.

Heitmeyer, M.E., Fredrickson, L.E., and Krause, G.F., 1991, Water relationships among wetland habitat types in the Mingo Swamp, Missouri: Wetlands, v. 11, p. 55-66.

Knowlton, M.F. and Jones, J.R., 1997, Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity: Wetlands, v. 17, p. 468-475.

Knowlton, M.F. and Jones, J.R., 2000, Seston, light, nutrients, and chlorophyll in the Lower Missouri River, 1994-1998: Journal of Freshwater Ecology, v. 15, p. 283-297.

Pflieger, W.L., 1997, The fishes of Missouri (rev. ed.): Jefferson City, Mo., Missouri Department of Conservation, 372 p.

Piper, R.G., McElwain, I.B., Orme, L.E., McCraren, J.P., Fowler, L.G., and Leonard, J.R., 1982, Fish hatchery management: Washington, D.C., U.S. Fish and Wildlife Service, 517 p.

Tibbs, J.E. and Galat, D.L., 1997, Larval, juvenile, and small fish use of scour basins connected to the Lower Missouri River--Final report to Missouri Department of Conservation: [Columbia, Mo], [USGS Missouri Cooperative Fish and Wildlife Research Unit], 133 p.

List of Tables and Figures

Tables

Figures

Table 2-1. List and description of Lisbon wetlands in which limnology and water quality was studied.

Wetland	Basin type/description	Primary water source	Permanence
2	Shallow basin, contains flooded willows and cottonwoods at high water stages	Mixed topflooding and stream influence	Temporary (4/16 > 6/15)
4	Deep scour	Topflooding, with some stream influence	Permanent
5	Scour, moderately deep, sedimenting rapidly	Topflooding	Persistent, considered permanent until it dried in the fall of 1999
8	Wide, shallow basin	Topflooding	Temporary (4/16 > 6/15)
9	Shallow basin, contains flooded willows and cottonwoods at high water stages	Mixed topflooding and stream influence	Ephemeral (first flooded 4/16, contained less than 6" water after 5/28)
10	Wide shallow basin, mostly wooded with young trees	Mixed topflooding and stream influence	Ephemeral (4-16 to 5-28)
11	Wide, very shallow marshy wetland, few trees but many wetland plants	Strongly stream influenced, at times with perceptible current, although flooded by the river during the flood.	Temporary (< 3/30 to > 6/15)
12	Moderately deep wetland formed by partial damming of an intermittent stream	Stream influenced, this wetland was never flooded by the river during the study	Persistent, considered permanent until it dried in August 1999
16	Deep scour	Topflooding from chute	Permanent
21	Moderately deep scour	Topflooding from chute	Temporary (4/16 > 6/15)
22	Shallow basin	Mixed backflooding, topflooding, and stream influences	Temporary
26	Very deep scour, steep sided	Backflooding from Cooper's Creek is primary connection to the river	Permanent
29	Narrow scour, usually less than one meter deep	Backflooding through a break in the levee is primary river connection	Temporary (4/16 > 6/15)

Table 2-2. Mean values of various water quality parameters in wetlands of Lisbon Bottom between late March and the end of July 1999. The apparent differences between the river and chute in some values (especially POC and total nitrogen) are the result of being unable to sample the river during high flows for safety reasons. River and chute water quality parameters were nearly identical on any given day when both measurements were taken.

	Wetland	Total Phosphorus (µg/L)	Total Nitrogen (mg/L)	Hardness mg/L	Alkalinity	POC[1]
Topflooding	4	212	0.665	131	286	357
	5	208	1.24	180	430	386
	8	241	1.24	179	433	667
	16	322	0.91	157	381	330
	21	408	0.87	205	496	468
Backflooding	26	316	0.97	160	387	382
	29	326	1.26	176	463	520
Mixed influence	2	181	1.30	162	347	526
	9	235	1.09	197	465	673
	10	320	1.66	175	401	394
	22	214	1.26	182	420	666
Stream influence	11	379	1.20	198	417	379
	12	309	0.44	165	351	283
River		358	1.94	232	629	612
Chute		386	2.82	239	609	854

[1]Particulate Organic Carbon

Table 2-3. Water quality in Lisbon Bottom wetlands during the 5/28 to 6/16 caged fathead minnow growth experiment. Values are means from each wetland. Wetlands 9 and 10, which dried during the experiment, are excluded.

Water source	Wetland	pH	Chlorophyll (μ/L)	Turbidity (ntu)	POC[1] (mg/L)	Temperature[2] °C
Topflooding	4	8.19	27.0	29.7	327	25.7
	5	7.95	52.3	42.8	356	25.8
	8	8.14	54.1	29.5	515	23.9
	16	8.51	36.8	22.5	392	25.1
	21	8.22	66.3	42.5	526	24.5
Backflooding	26	8.43	93.1	38.0	339	24.7
	29	8.04	64.5	29.5	450	25.7
Stream-influenced	11	8.00	31.1	42.0	302	25.6[3]
	12	7.96	13.1	23.3	242	26.3[3]
Mixed influences	2	8.22	54.3	36.0	644	20.5[3]
	22	8.11	69.1	43.3	728	26.2[3]

[1]particulate organic carbon
[2]from *in situ* temperature loggers
[3]data is mean of last three days of study, due to equipment malfunction

Table 2-4. Mean number of zooplankton organisms per liter in Lisbon Bottom wetlands during the fathead minnow growth study. Wetlands 9 and 10 are not included because they dried during the study. Rotifers may not have been adequately sampled because the mesh size was larger than that normally used for the capture of rotifers.

	Wetland	Cladocerans	Copepods	Rotifers	Total Organisms
Topflooding	4	15.6	8.7	4.7	28.9
	5	305.0	41.8	59.5	406.4
	8	553. 8	168. 9	170.3	892.9
	16	1293.1	849.6	5.3	2148.0
	21	114.2	60.7	0.0	175.0
Backflooding	26	165.8	519.6	0.7	686.0
	29	187.7	149.8	33.7	371.2
Mixed influence	2	221.6	96.1	178.9	496.7
	22	29.8	41.1	2.0	72.9
Stream influence	11	2.7	5.3	0.0	8.0
	12	26.2	21.6	1.1	48.9

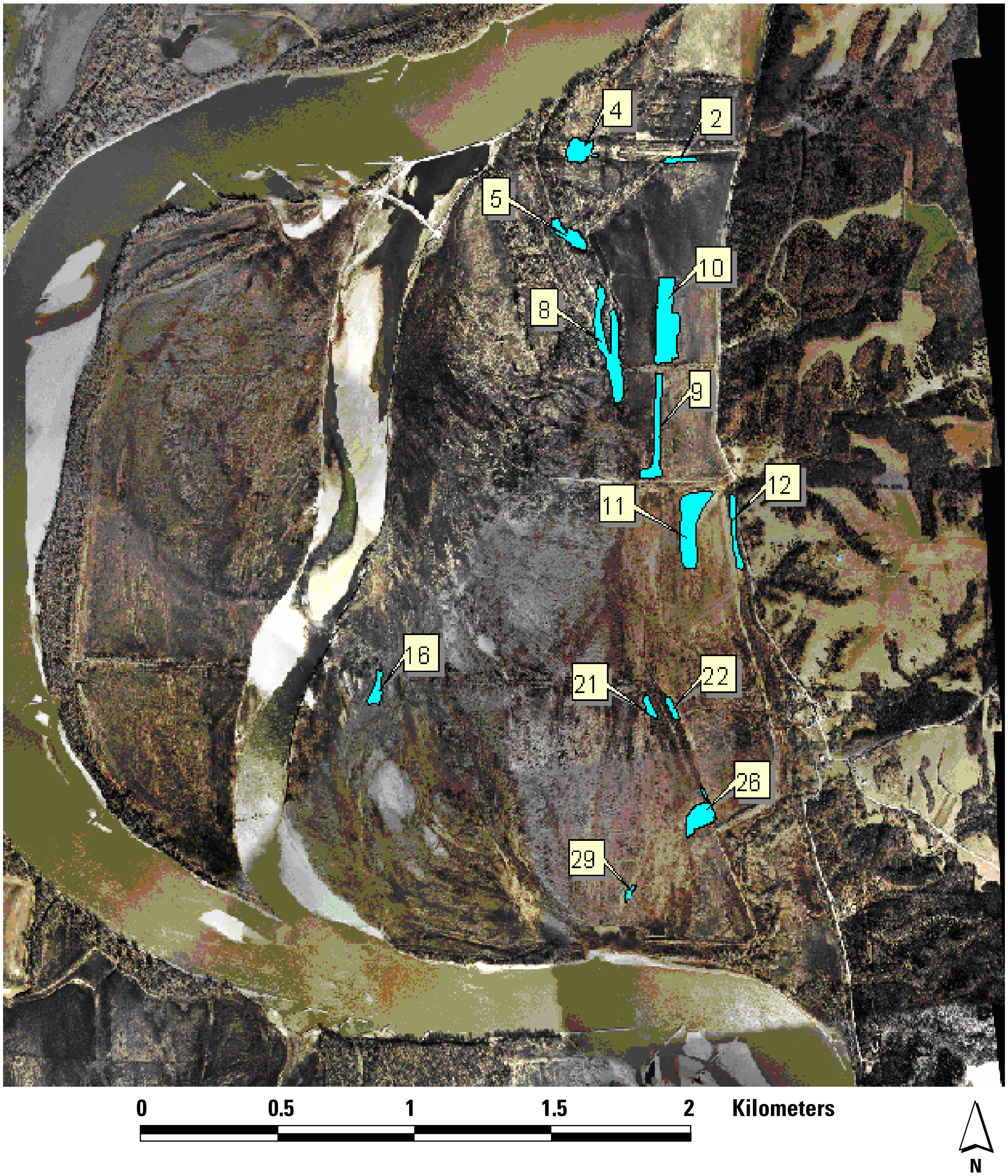

Figure 2-1. Map of Lisbon Bottom with numbered wetlands where limnological measurements were made. *Background photo courtesy of U.S. Army Corps of Engineers, Kansas City, MO, March 2000.*

A

B

Figure 2-2. Two pictures taken in the same area (Wetland 22), showing normal and flooded appearances. The tree in the left of figure A is the tree to the left in figure B. Picture B was taken from a canoe. The cottonwood sapling protruding from the water in the foreground of figure B was approximately seven feet tall.

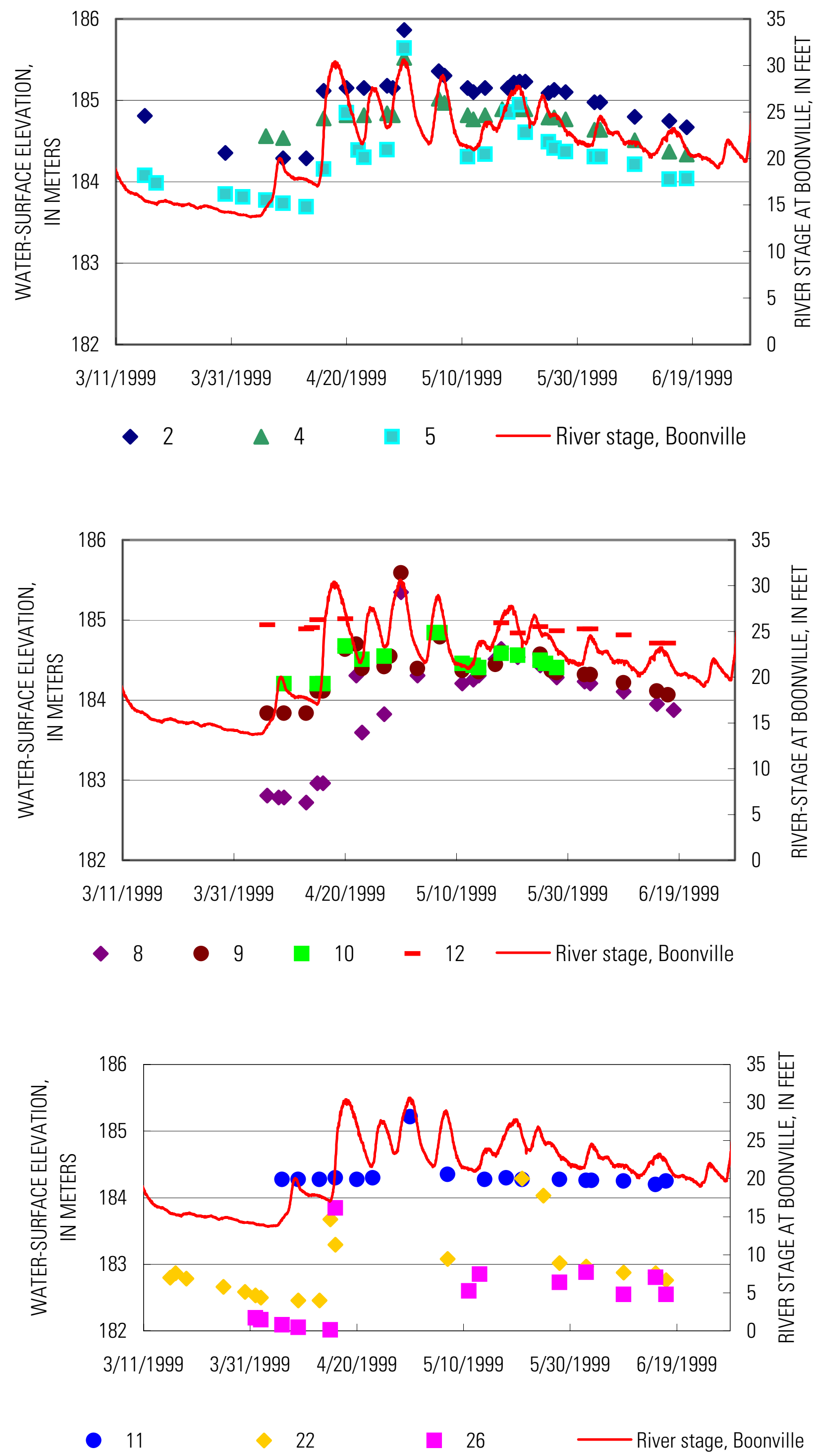

Figure 2-3. Surface elevations of wetlands at Lisbon Bottom, March–June 1999. Note that, with the exception of the measurements made by canoe on April 30, these data do not capture periods of greatest flood, when the wetlands were inaccessible to walkers.

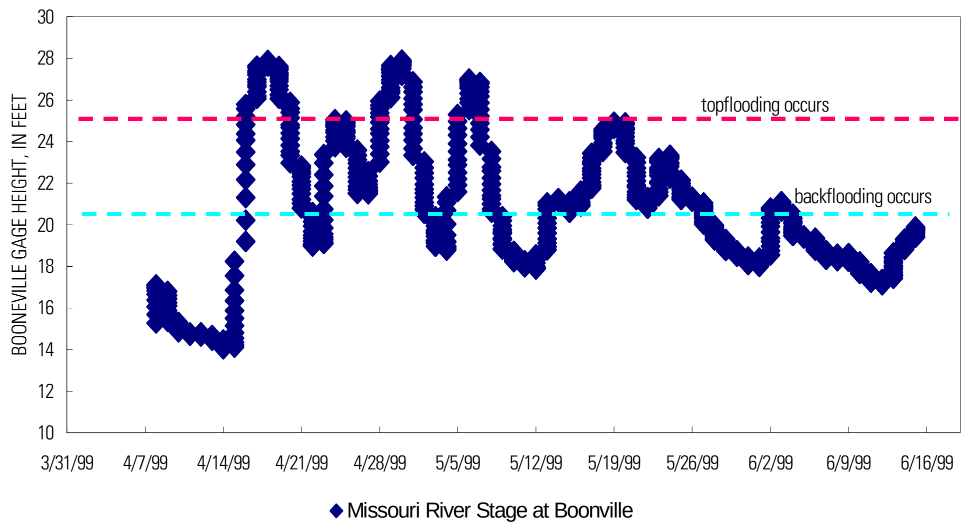

Figure 2-4. Missouri River stage at Boonville. The red horizontal line indicates the gage height (25.5 ft) at which water was seen to enter Lisbon Bottom through a crevasse that passes through the levee, topflooding Wetlands 4 and 5 and other portions of the northern bottom. Wetland 26 is believed to have connected to the river via backflooding through a narrow channel at approximately 20.5 feet on the Boonville gage (light blue horizontal line).

Figure 2-5. Unconsolidated sediment freshly deposited after a flood of Lisbon Bottom.

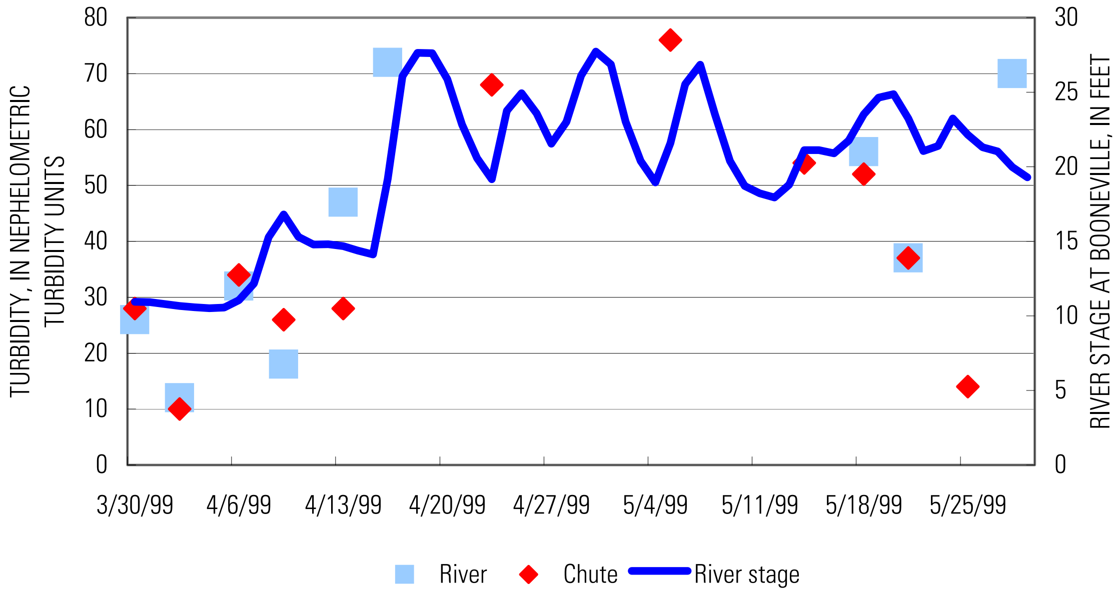

Figure 2-6. Turbidity of the Missouri River and the Lisbon Chute. Turbidity of the river and chute were very similar and both rose during the period of flooding. There are many missing values, especially from the river site, because safety concerns prohibited many collections during periods of high flow.

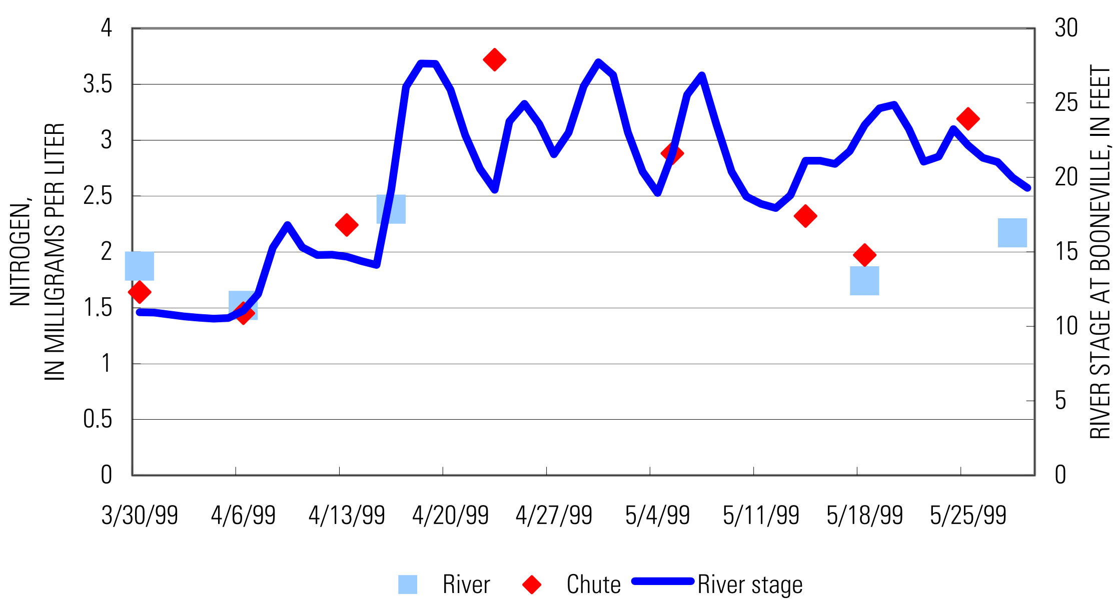

Figure 2-7. Nitrogen concentrations of the Missouri River and the Lisbon Chute. Nitrogen concentrations of the river and chute were very similar and were correlated with river stage. There are many missing values, especially from the river site, because safety concerns prohibited many collections during periods of high flow.

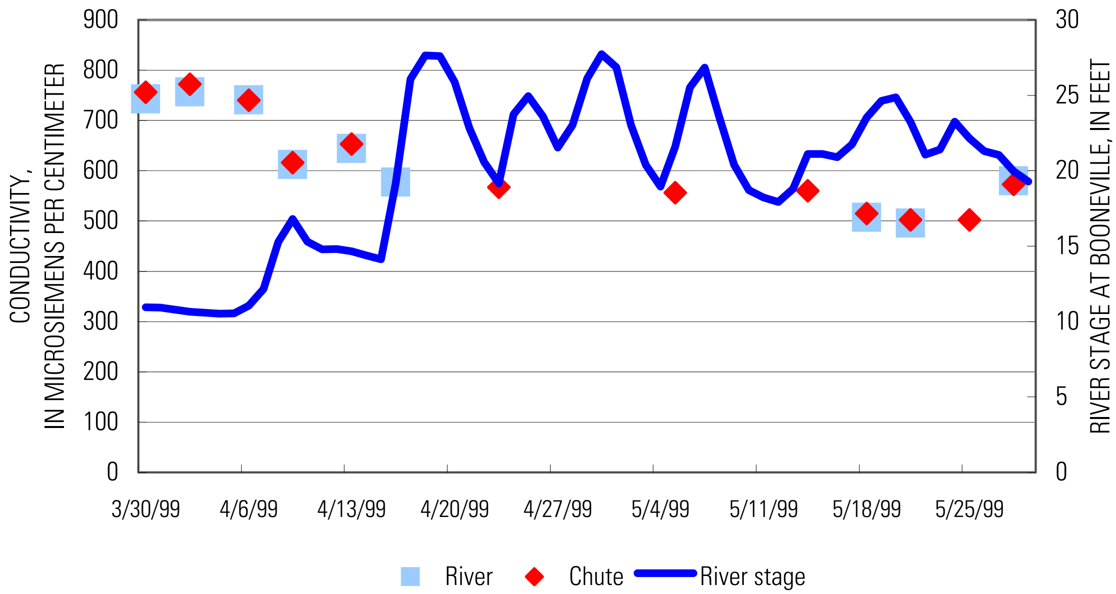

Figure 2-8. Conductivity of the Missouri River and the Lisbon Chute. Conductivity of the river and chute were nearly identical and were inversely correlated with river stage. There are many missing values, especially from the river site, because safety concerns prohibited many collections during periods of high flow.

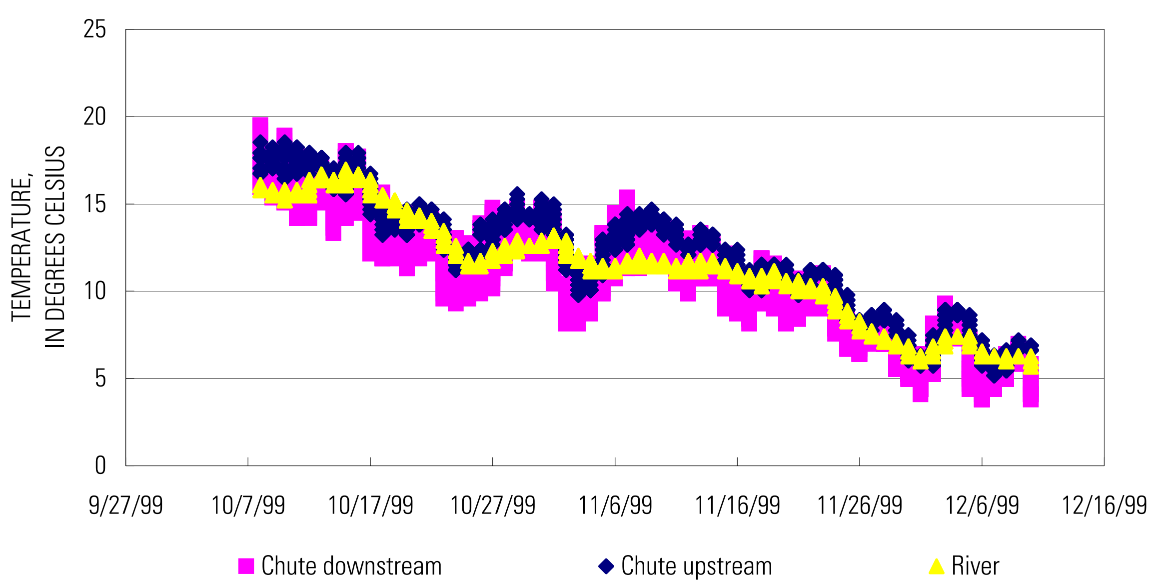

Figure 2-9. Temperature of the Missouri River and two sites within the Lisbon Chute. The temperature logger in the Missouri River was lost during the spring 1999 study; these data were collected in the fall of 1999. However, the data do indicate that the chute has less thermal inertia than the river and that diurnal fluctuations in chute temperature do occur, at least during periods of lower flow.

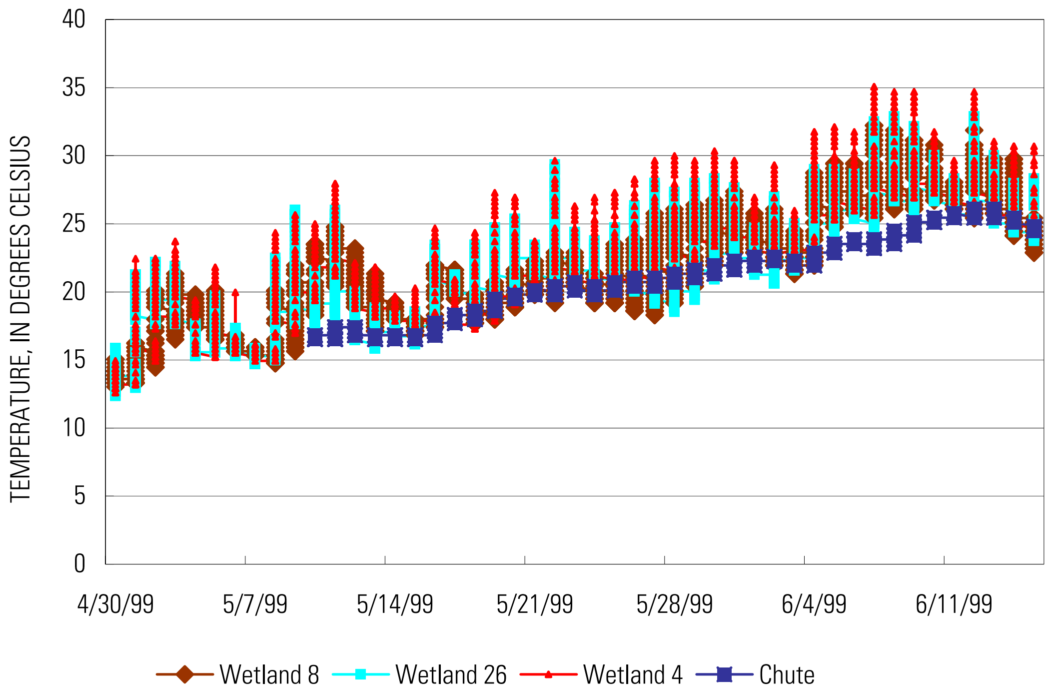

Figure 2-10. Temperature logger data from selected Lisbon Bottom wetlands and the chute. Loggers were installed 10 cm below the surface. Note that daily temperature variance was much higher in the shallow wetlands than in the chute.

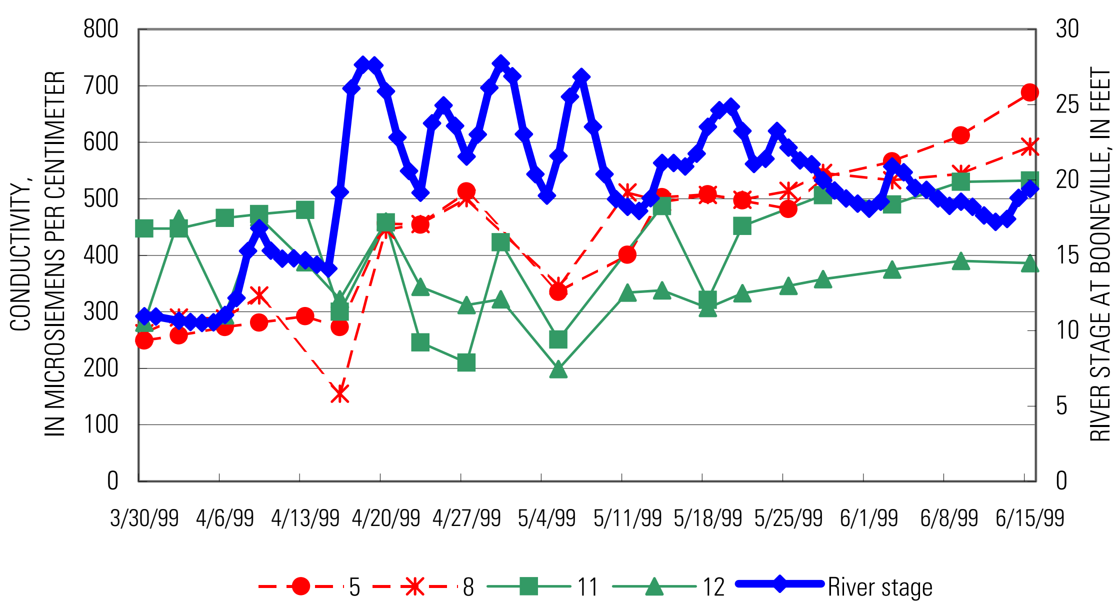

Figure 2-11. Conductivities of stream-influenced wetlands (in brown) and topflooding wetlands (in red) in relation to river stage. Note that conductivities of the top flooding wetlands increase after flooding by the river. Stream-influenced wetlands varied dramatically in response to local rain events, but were generally lower than river-influenced wetlands after flooding from the river.

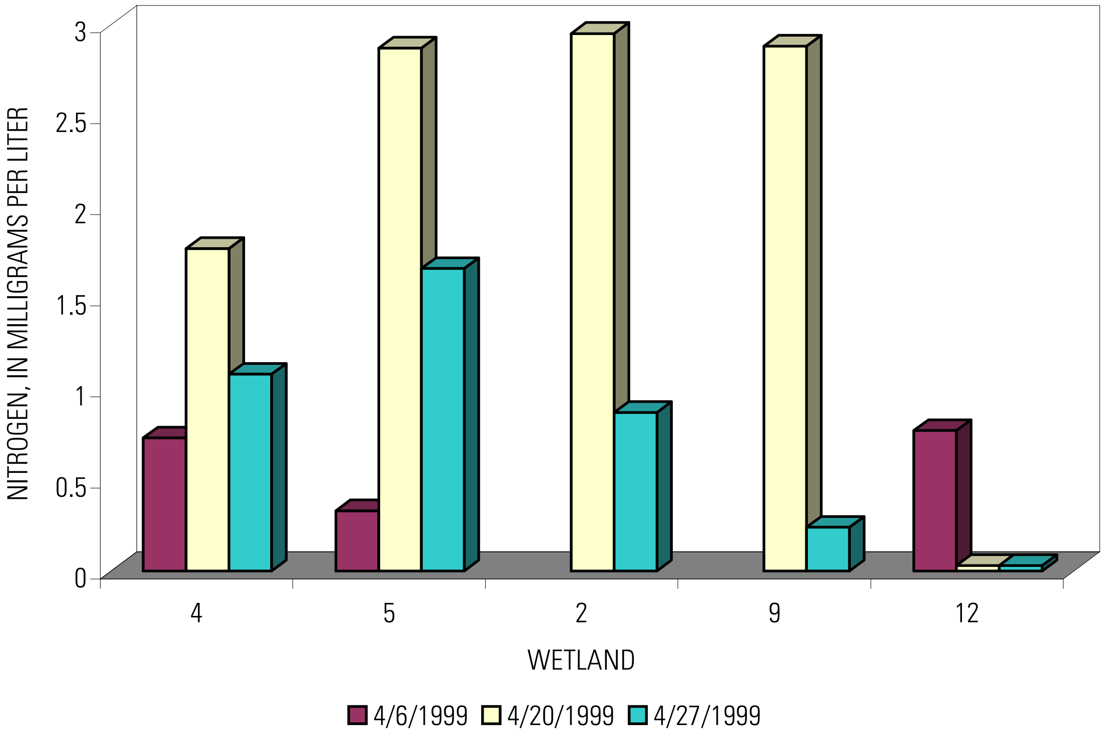

Figure 2-12. Total nitrogen concentrations in selected Lisbon Bottom wetlands before, during, and after a flood by the river. Wetlands 4 and 5 are topflooding scours. Wetlands 2 and 9 were shallow wetlands that contained no water prior to flooding. Wetland 12 was not inundated by the river. Wetlands 4, 5, 2, and 9 are in order of decreasing mean depth on 4/27/99.

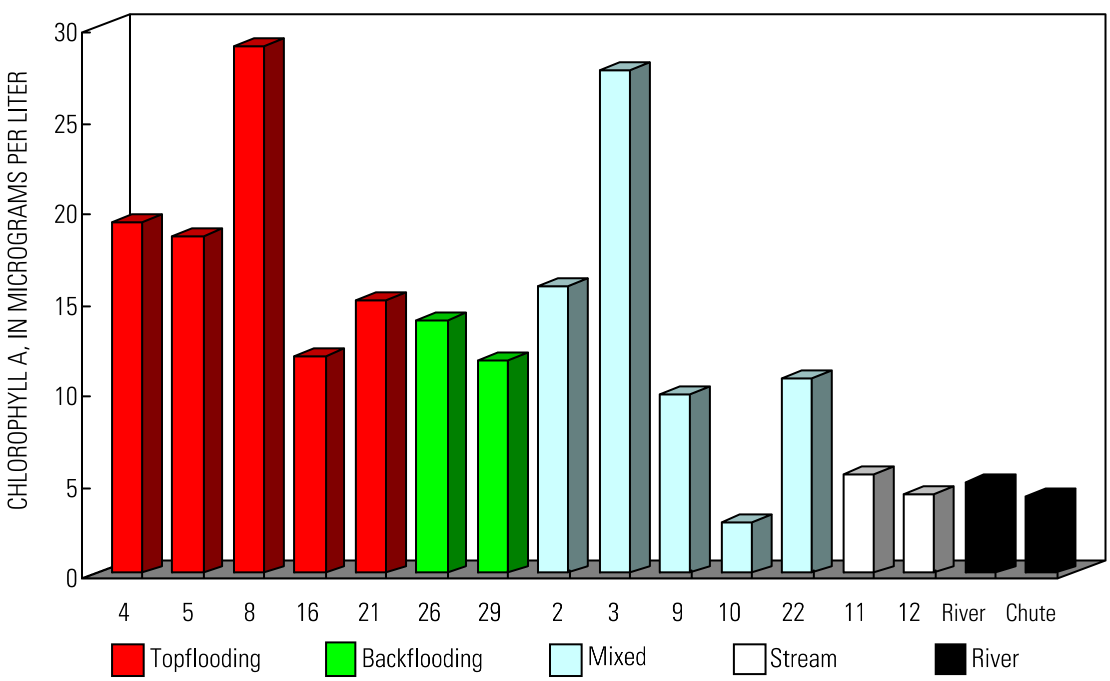

Figure 2-13. Mean chlorophyll *a* concentrations in Lisbon Bottom wetlands between March 30 and June 15, 1999.

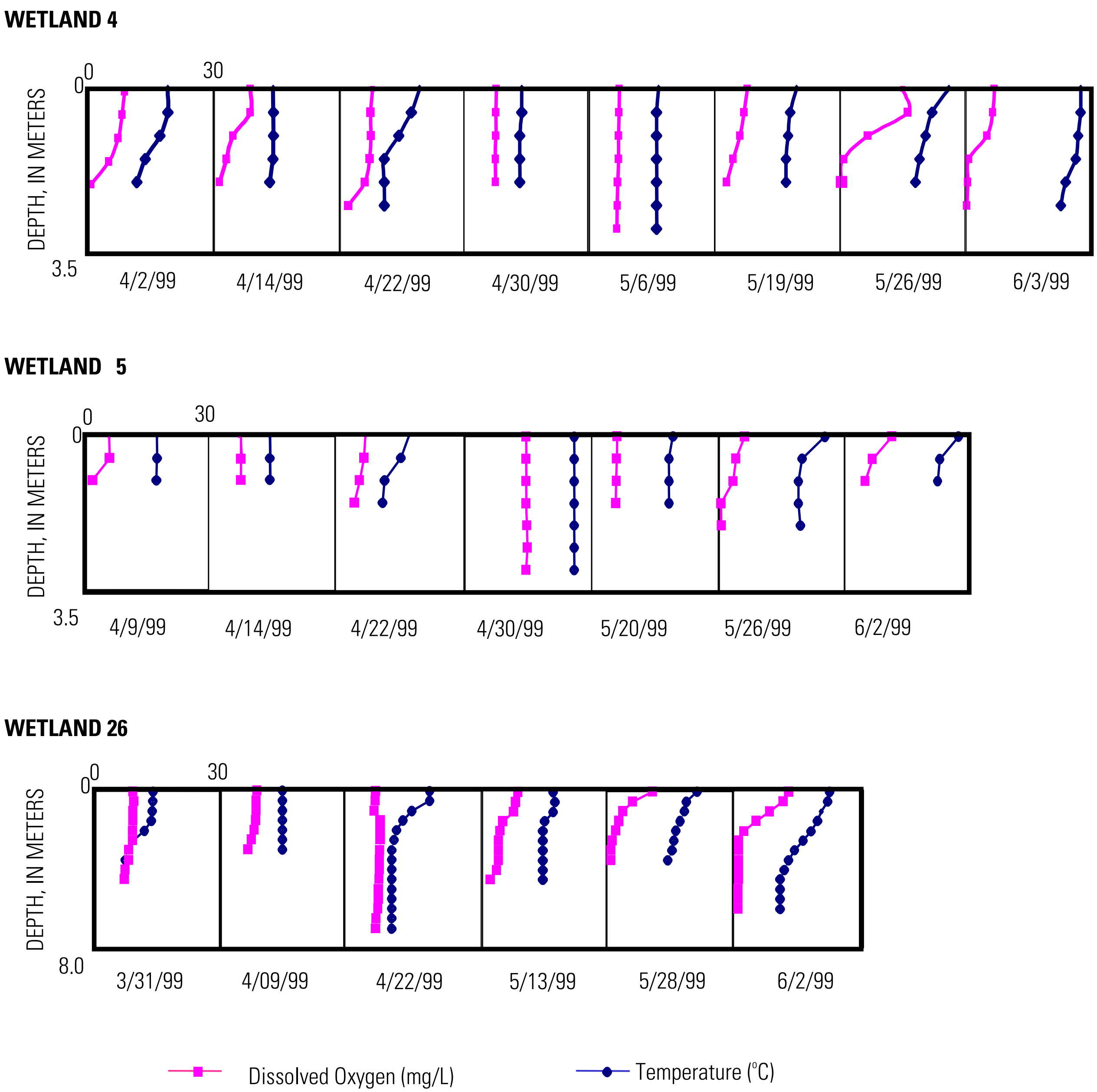

Figure 2-14. Temperature and dissolved oxygen profiles in three scour wetlands at Lisbon Bottom. Each square box is one sampling event. Surface values are at the top of the box; measurements are taken at half-meter depth intervals. These figures show data from the deepest point in each wetland. River water was passing through the crevasse April 16 to April 20, April 28 to May 1, May 5 to May 7, and May 19 to 20.

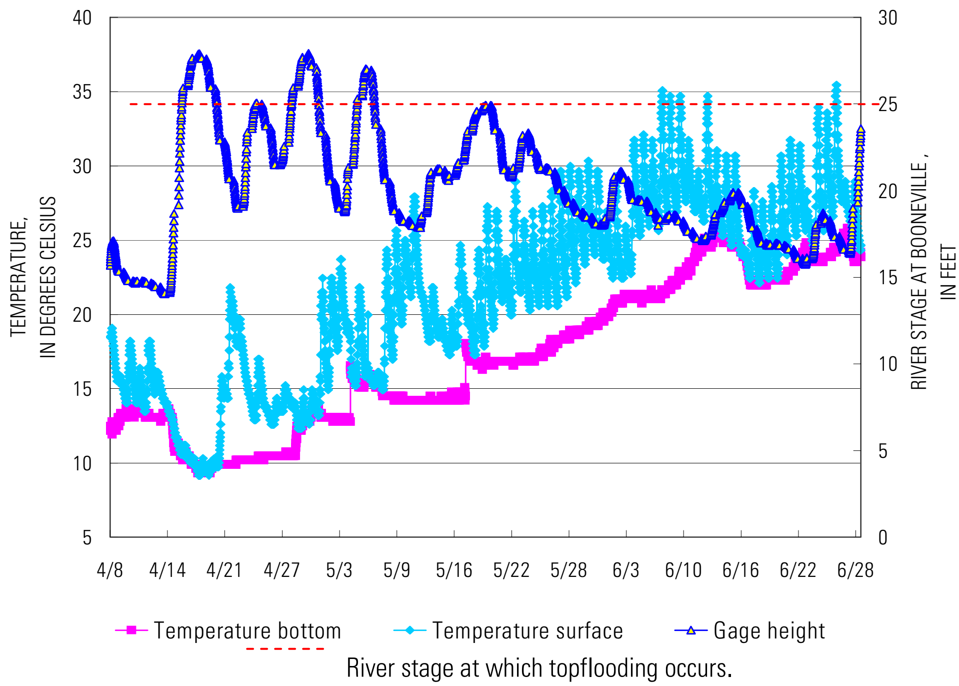

Figure 2-15. Temperature logger data from the surface and bottom of topflooding Wetland 4, and Boonville river gage height. Note that when water passes through the notch, surface and bottom temperatures are equivalent, but stratification redevelops quickly.

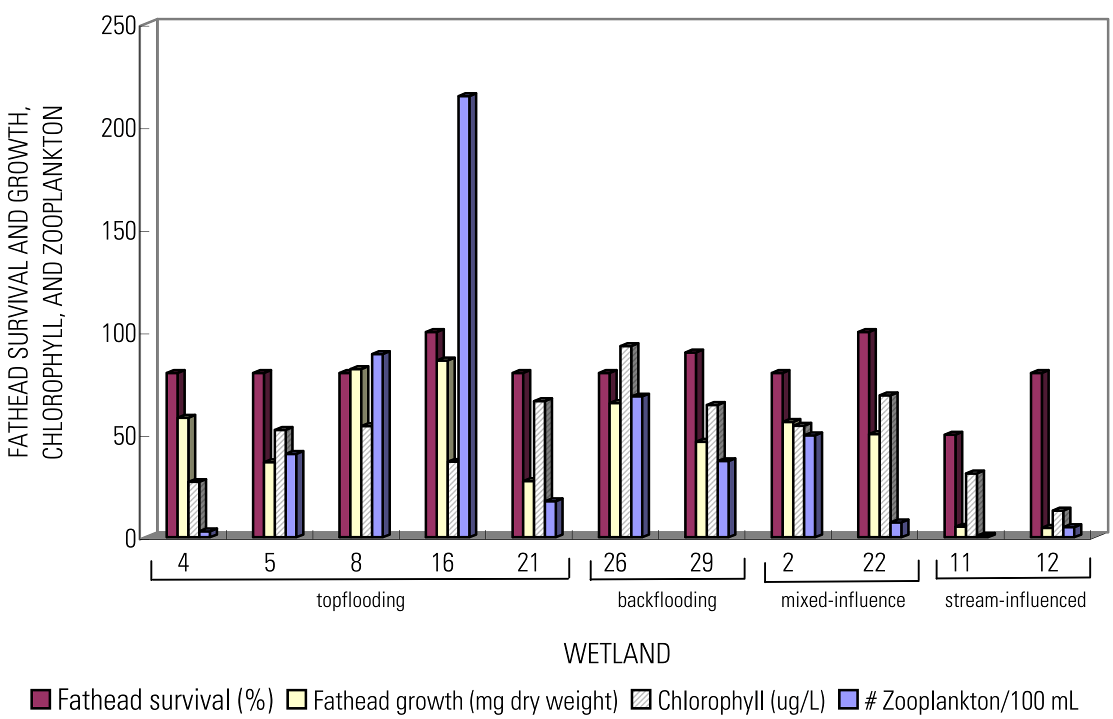

Figure 2-16. Survival and growth of caged fathead minnows in Lisbon Bottom wetlands with chlorophyll *a* concentrations and mean number of zooplankton per 100 mL water.

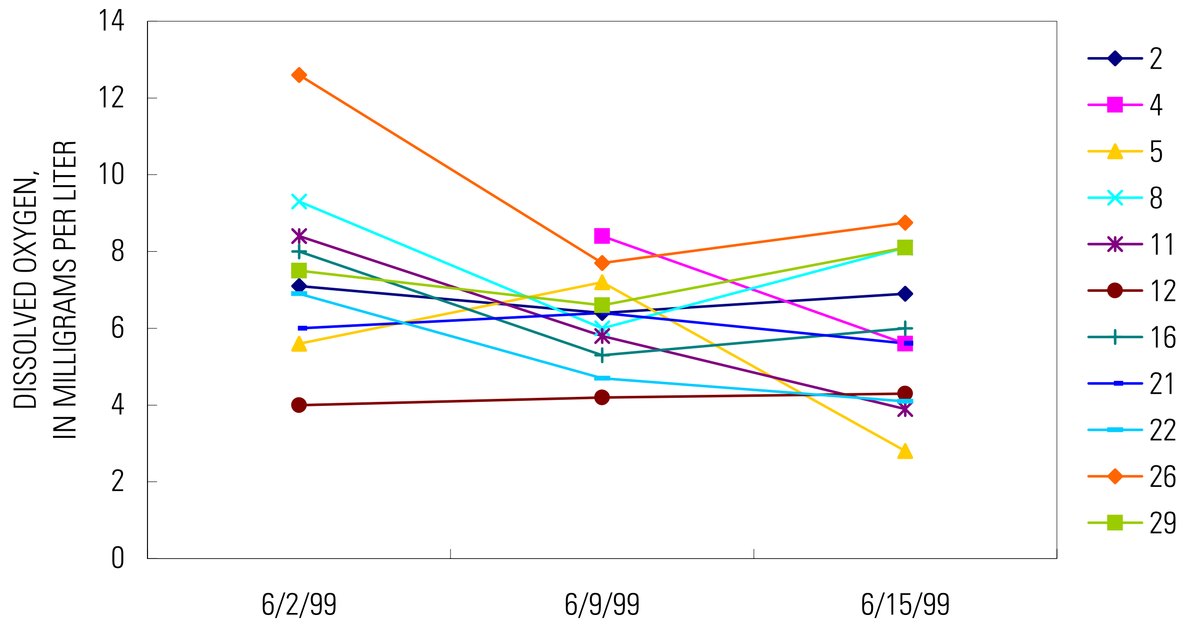

Figure 2-17. Dissolved oxygen in Lisbon Bottom wetlands during the fathead minnow experiment (5/27/99–6/16/99). Measurements were made adjacent to the floating cages.